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Life cycle inventory of recycling portable nickel–cadmium batteries

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Abstract

In this study, the environmental impact of recycling portable nickel–cadmium (NiCd) batteries in Sweden is evaluated. A life cycle assessment approach was used to identify life cycle activities with significant impact, the influence of different recycling rates and different time boundaries for emissions of landfilled metals. Excluding the user phase of the battery, 65% of the primary energy is used in the manufacture of batteries while 32% is used in the production of raw materials. Metal emissions from batteries to water originate (96–98%) from landfilling and incineration. The transportation distance for the collection of batteries has no significant influence on energy use and emissions. Batteries manufactured with recycled nickel and cadmium instead of virgin metals have 16% lower primary energy use. Recycled cadmium and nickel metal require 46 and 75% less primary energy, respectively, compared with extraction and refining of virgin metal. Considering an infinite time perspective, the potential metal emissions are 300–400 times greater than during the initial 100 years. From an environmental perspective, the optimum recycling rate for NiCd batteries tends to be close to 100%. It may be difficult to introduce effective incitements to increase the battery collection rate. Cadmium should be used in products that are likely to be collected at the end of their life, otherwise collection and subsequent safe storage in concentrated form seems to offer the best solution to avoid dissipative losses.

Keywords: Nickel–cadmium batteries; Recycling; Life cycle assessment; Materials management

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1. Introduction

Portable nickel–cadmium (NiCd) batteries have been used in electronic products for many decades. In 1990, cadmium from portable NiCd batteries contributed 60% (84 ton Cd) of the inflow of cadmium in products to the Swedish society (Bergbäck, 1992). The use of portable NiCd batteries has decreased in Sweden during recent years but cadmium in portable and industrial NiCd batteries still contributes 90% (41 ton) of the inflow of cadmium-containing products in Sweden in 1998 (Olsson, 2000). The market for portable electronic products is expanding and the global production of portable rechargeable batteries grew at an annual rate of 14% during the past 10 years (Fig. 1). The total number of portable cells produced in 1999 was 2.9×10^9 cells (excluding $\approx 0.3 \times 10^9$ small sealed lead–acid cells). The introduction of new types of electrochemical systems (nickel–metal hydride, lithium-ion and lithium–polymer batteries) has decreased the global market share of NiCd batteries from almost 100% in 1992 to 49% in 1999. NiCd battery production in Japan, accounting for 42% of NiCd batteries manufactured globally 1998, increased by 29% during 1990–1994, but later decreased by 33% during 1994–1998 (Fujimoto, 1999).

In September 1997, the European Commission called for a ban on portable NiCd batteries to be introduced in 2008 because of poor recovery rates. Stakeholders (EPBA, 1999; Sempels, 1999; CollectNiCad, 2001) claim that the economic effects will be unacceptable and increased collection and reprocessing of portable NiCd batteries may reduce the environmental effects of NiCd batteries significantly.

In 1998, the Swedish battery ordinance came into effect, which states that all kinds of household batteries must be collected to avoid the spread of cadmium, mercury and lead. In 1998, approximately 1400 ton of mixed household batteries were collected in Sweden (Olsson, 2000). The only types of batteries sent for

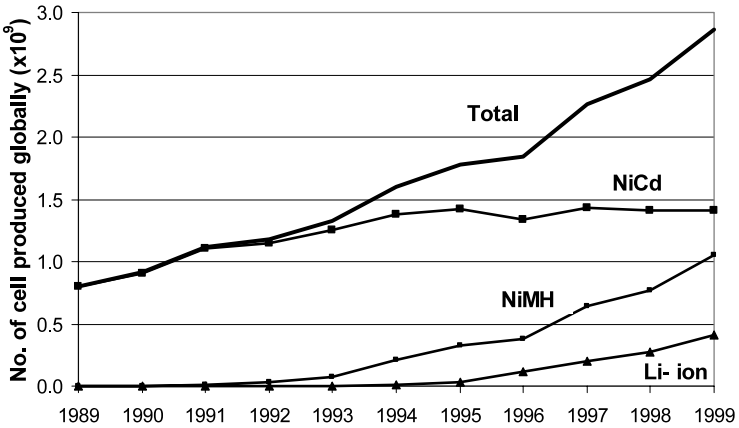

Fig. 1. Number of rechargeable portable battery cells produced globally from 1989 to 1999 (portable lead–acid batteries not included) (Fujimoto, 1999; Noréus, 2000). (NiCd, nickel cadmium, NiMH, nickel–metal hydride, Li-ion, lithium ion).

materials recovery are NiCd (8 wt.% of the amount collected), small lead–acid batteries (3 wt.%) and nickel–metal hydride (1 wt.%), while the rest is landfilled (Renova, 1999).

Improved material management can lead to better utilization of refined materials, decreased use of primary materials and energy resources and a reduced need for landfill areas. The benefits of recycling materials from an economic, environmental and technical point of view depend on many parameters, such as transport distance, recycling processes and type of material. Louis et al. (1998) reviewed 314 articles on studies of the economics and environmental costs and benefits of recycling post-consumer waste. They concluded that there were major gaps in the literature regarding the assessment of the environmental costs of recycling, and no analyses supported any particular target level of recycling.

Recycling of NiCd batteries is interesting, since cadmium has been a political issue for many years, and the substance is toxic and geologically scarce. One of the proposed criteria for the EU environmental labelling of batteries (Baumann and Scholl, 1997) requires the importer of batteries to join a battery-recycling programme. However, only a few studies have quantified the benefits of battery collection and the recovery of metals (Hofstetter and Häne, 1990; Törnblom, 1996; Lankey, 1998; Karlsson, 1999). Conclusions from an evaluation of mature NiCd battery technology may be used to assess emerging new electrochemical systems.

2. Aim and scope

The aim of this study was to assess the environmental effects of recycling portable NiCd batteries in Sweden and to identify life cycle activities with significant potential environmental impact. The sensitivity of the NiCd battery system has been evaluated by varying recycling rates and emission factors for landfilled metals. Materials management of cadmium is discussed based on the results obtained.

The assessment was made using a life cycle inventory (LCI), which includes compiling an inventory of environmentally relevant inputs and outputs related to the functionality of a product (ISO, 1997). When using LCI methodology increased recycling normally will decrease energy use and metal emissions for most kinds of product systems. The contribution of this study is to quantify these benefits for a NiCd battery system with static modelling.

LCI was chosen as environmental analytical tool, because it includes if decreased emissions are shifted to other environmental problems and compared to other methods (e.g. material flow analysis, cumulative energy requirements analysis) it includes potential environmental impact connected to the material and energy flows.

Environmental impact related to metal emissions and energy use are regarded as the most significant activities for the batteries (Rydh, 2001). Environmental impact assessment was not performed since the environmental impact is strongly connected to these activities.

The functional unit of the study was defined as 'a battery with an energy storage capacity of 1.0 W h electrical energy'. This corresponds to a cylindrical NiCd battery with a mass of 25 g (40 W h/kg).

Emissions and resource use in the user phase of the battery were excluded from the study since these do not influence the materials management of metals for the functional unit chosen. Various kinds of end-of-life treatment (recycling, landfilling and incineration) were considered. When possible, allocation procedures proposed by ISO, (1998) where applied to distribute environmental impact from activities with multiple outputs. Closed-loop recycling was assumed for cadmium and nickel.

It was assumed that the NiCd batteries were manufactured in Germany and used in Sweden. Data on raw materials extraction and refining (excluding capital and personnel) from cradle to gate are based on average data from manufacturers. Average transportation distances are estimated for materials production, collection and recycling of batteries in Sweden. Emissions from electricity generation (extraction, refining and combustion of fuels) were calculated for base case based on a country specific mix for electricity generation. The sensitivity of the model was evaluated by comparing the base case with hydro and coal power electricity generation, respectively.

Metal emissions from landfills to water are modelled for a surveyable time corresponding to 100 years, and in a long-term perspective, representing a worst case scenario or hypothetical infinite time when all metals have been completely released to the environment (Finnveden et al., 1995).

Inventory data were compiled from LCA databases, literature, interviews and reports on NiCd batteries (Kertes, 1996; Jensen and Petersen, 1999). Information on collection and sorting of batteries by municipalities was acquired through questionnaires.

3. Inventory

3.1. Manufacturing

Inventory data used for the calculations are presented in Rydh and Karlström (2001). The most common sizes of rechargeable portable battery cells are AA and Sub C cells (weight ≈ 22 and 50 g, respectively). Table 1 shows the material requirements corresponding to the functional unit of 1 W h, almost equivalent to a cell of AA size (average voltage = 1.2 V, capacity = 0.6–0.8 A h).

The average primary energy use for extraction and refining of cadmium (from zinc mining) and nickel has been estimated to be 70 MJ/kg Cd (Boustedt and Dove, 1998) and 159 MJ/kg Ni (Kirmayer, 1995), respectively. The primary energy requirements for manufacturing processes of batteries produced in Hagen, Germany were calculated to be 140 MJ/kg battery (Kirmayer, 1995). Metal emissions were approximated from the manufacture of stationary NiCd batteries in Sweden.

Table 1
Material requirements for a 1 W h portable nickel–cadmium battery (g)

Components	Materials											
	Steel	Ni	Cd	Hydroxides	H_2O (demin.)	KOH (30%)	PA	PVC	Co	Rubber[b]	Sum	Wt.%
Negative electrode	0.77	1.16	3.88	0.29							6.10	24.4
Positive electrode	1.65	2.93	0.21	1.74					0.12[a]		6.65	26.6
Electrolyte					2.00	0.86					2.87	11.5
Separator							0.53				0.53	2.1
Can, header, cover	7.38										7.38	29.5
Vent										0.06	0.06	0.3
Current collector	1.02										1.02	4.1
Label								0.26			0.26	1.1
Gasket							0.13				0.13	0.5
Sum	9.80	5.11	4.09	2.03	2.00	0.86	0.65	0.26	0.12	0.06	25.00	100.0
Wt.%	39.2	20.5	16.4	8.1	8.0	3.5	2.6	1.1	0.5	0.3	100.0	

Source: processed data Kirmayer (1995).
[a] Not taken into account.
[b] Assumed to by polypropylene.

3.2. Collection and recycling

The routes for final disposal are not well known for portable NiCd batteries since the in- and out-flows from the technosphere are difficult to quantify. It is uncertain how many NiCd cells enter the market since they may be assembled in packs containing variable numbers of cells. Given that NiCd batteries are used in power packs or are integrated in equipment, it is probable that large amounts of batteries are still being stockpiled and that steady state between in- and out-put has not yet been reached (Edwards and Schelling, 1999). It is thus uncertain where accumulated batteries will end up in the future. The registered annual supply of NiCd batteries to Sweden during the period 1990–1995 was 450–635 ton (mean 547 ton) (SCB, 2000). An average of 142 ton of NiCd batteries were collected annually from 1996–1999. Assuming a lifetime for NiCd batteries of 6–9 years gives a collection rate of 22–31% (mean 26%). Recent collection results indicate, however, that the amount of batteries collected is increasing. Another way to determine the recycling rate is to compare the amount of batteries collected with the amount found in municipal solid waste (MSW). However, sorting studies of mixed MSW have been carried out on a few samples only, which makes them less reliable. In the base case, it is assumed that end-of-life pathways for NiCd batteries are distributed between recycling (25%) and MSW. Of the MSW fraction, 60% is incinerated and 40% is landfilled. The fraction of batteries lost directly to the environment is not taken into account as the amount is so small (Rydh, 1999).

Modelling of consumer car transportation to recycling sites and local truck transports of batteries were adapted from a study of glass collection (Edwards and Schelling, 1999). The model considers that the recovery rate increases with increasing recycling site densities. Shorter distances per kg recovered battery have to be driven by private cars at higher site densities, thus leading to a minimum at recovery rates greater than 90%. The transportation distances involved in collecting mixed household batteries from battery collection boxes and taking them to a central point within a municipality vary in the range 30–250 km (average 100 km) for the different municipalities in Sweden. Batteries are often transported together with other waste. The fuel consumption per kg recovered battery for local truck transports is considered to be constant for recovery rates 10–85% (Edwards and Schelling, 1999). At higher recovery rates, the fuel consumption increases rapidly due to longer distances to cover all sites and the decreasing amount of material available per site. Energy use for production of battery collection boxes was excluded since it was found to be insignificant.

The municipalities are responsible for sorting the batteries into different fractions. Twenty-five percent of the portable batteries collected annually in Sweden is sent to Gothenburg for sorting in a semi-automatic sorting machine (Renova, 1999). Since NiCd and small lead–acid batteries are the reason for the collection of household batteries, the environmental impact of transportation of all household batteries is allocated to these types of batteries. The fraction of NiCd batteries (8 wt.%) of all household batteries collected is transported an average distance of 600 km to AB SAFT in Oskarshamn, Sweden for cadmium recovery.

In 1998, 143 ton portable NiCd batteries, corresponding to 19 ton of cadmium, were recycled at AB SAFT, Oskarshamn, Sweden (SAFT, 1983–1998). Recoverable materials (76 wt.% of NiCd battery) are cadmium and nickel-iron scrap. The cadmium recovered is used in the production of new industrial NiCd batteries at SAFT, and nickel-iron scrap is sent to smelters for use as alloying metal in the steel industry. However, in this study, it is assumed that the cadmium recovered is used in the production of new portable batteries to avoid the use of different allocation procedures, which must be applied when recycling materials in cascade.

Heat energy is recovered from plastics (4 wt.% of NiCd batteries). The input of heat energy to the recycling process is recovered in the production of positive active material (drying of nickel hydroxide) in another process in the factory. Allocation of energy between the two product systems was made by system expansion (ISO, 1997), where the energy recovered from battery recycling is assumed to replace liquid petroleum gas which would otherwise have to be used at the plant if the recycling process had not been equipped with a heat exchanger.

3.3. Waste incineration and landfilling

In Sweden, approximately 60% of MSW (excluding coarse waste) is incinerated in waste incineration plants. The proportion of MSW incinerated is expected to increase in the future due to higher fees for the deposition of combustible waste. MSW incineration plants in Sweden are equipped with extensive air and water cleaning equipment to capture pollutants. However, small amounts of metals bound to aerosols may escape through filters. Metals are stored in bottom and flue ash. Organic sulphides are added to the ash for complexation of metals, and lime is added to maintain an alkaline environment. The stabilised sludge is landfilled at municipal sanitary landfills. Forty percent of the MSW is landfilled directly, which means that almost all metals entering the waste stream end up in landfills, if the metals are not recovered from the ash. Depending on the time perspective chosen, different amounts of metals will leached out into the environment giving rise to potential environmental impact. The rate of degradation and mobilisation of metals from batteries is very complex and depends on battery-specific parameters (type of casing and state of charge) as well as site-specific parameters (e.g. pH, redox potential, amount of oxidising agents, cation-exchange capacity, organic content, solid-to-liquid ratio and soil texture) (Jones et al., 1978). In degradation tests of NiCd batteries in landfills, Oda, 1989 concluded that cadmium from whole batteries would not be released for two to four decades in a landfill subjected to normal rainfall.

A crucial methodological issue in life cycle assessment is determining the system boundary for the cradle and the grave of a product system. Finnveden (1996) suggested that both a short- and long-term perspective should be considered. The short-term scenario is a surveyable time, which is the later part of the methane stage until a pseudo-steady-state has been reached. This period is of the magnitude of one century in northern Europe, but may be shorter in countries with higher average temperatures. Emission factors are calculated by considering the concentrations of

metals in the leachate from landfills, the amount of leachate, and the metals in waste. In a short-term perspective the emission of metals from batteries may be very low.

Large amounts of heavy metals stored in landfills pose a potential threat in a long-term perspective. In this study, the long-term scenario is a hypothetical, infinite period, which is the period from landfilling until the landfilled material has been completely released to the environment and has become part of the biosphere (Finnveden, 1996). It is assumed that landfilled metals are corroded but immobilised as solid compounds, or in some cases as ions, adsorbed in the waste matrix or in the biomass, which prevents them from being biologically available. However, due to the laws of equilibrium, these products will be continuously leached out by percolation. Landfill erosion, fire or flooding may change the environmental conditions and cause a major release of metals.

The emission factors for a surveyable time are very small compared with the long-term perspective (landfilled metal to water: 5×10^{-3} kg Ni/kg Ni and 5×10^{-4} kg Cd/kg Cd) (Finnveden, 1996). Although metal emission factors found in the literature vary by several orders of magnitude, the values indicate that it will take thousands of years to mobilise and disperse accumulated metals.

4. Results

4.1. Activities with potential environmental impact

The following results represent the assessment of the current situation regarding battery handling in Sweden where 25% of NiCd batteries are recycled, 45% incinerated and 30% landfilled. Table 2 shows that 3.1% of the total primary energy is based on renewable energy, 65% of the primary energy is used in battery manufacture and 32% is used in the production of raw materials. The processes of electrolysis of the negative electrode and impregnation of the positive electrode contribute significantly to the energy use in the manufacturing phase. Of the total energy, 0.9% is used in battery distribution, collection and sorting.

Fifty-five percent of the CO_2 emission originates from battery manufacturing, 44% from raw materials production and 0.8% from battery distribution, collection and sorting. Transportation in the materials production phase contributes 1% to the total CO_2 emission.

Fifty-six percent of the NO_x emission originates from materials production. Collection and sorting contribute 10% to this emission since short-distance transportation gives rise to high NO_x emission and the transported mass of batteries is high since all the household batteries collected are included. Assuming doubled transportation distances for collection and recycling increases the total NO_x emission by 4% while energy use and other emission change insignificantly.

Consumption of (metal) resources is ascribed to unit processes, which disperse resources (concentrated materials) and make them difficult to recover. Consequently, the greatest resource use is found in the incineration and landfilling activities.

Table 2
Selected inventory data for NiCd battery life cycle (excl. user phase) specified for different activities

	Total	Raw mat'l (excl. Cd) (%)	Cd extraction/refining (%)	Battery manufact. (%)	Collection and sorting (%)	Recycling process (%)	Incineration and landfill (%)
Renewable energy (MJ/W h)	0.16	33.3	8.6	49.1	0	8.5	0.5
Non-renew. energy (MJ/W h)	4.92	26.8	4.8	66.0	0.9	1.3	0.1
CO_2 [a] (kg/W h)	0.37	39.8	4.1	55.2	0.8	−0.2	0.3
NO_x (g/W h)	0.50	33.6	22.0	30.8	10.2	3.1	0.0
SO_x (g/W h)	4.17	92.6	2.7	4.6	0	−0.1	0
Cd (resource) (g/W h)	3.1	0	0	0	0	0	100.0
Ni (resource) (g/W h)	3.8	0	0	0	0	0	100.0

End-of-life: 25% battery recycling; 30% landfill and 45% incineration; short-term metal emission factors from landfills.
[a] Carbon from fossil sources.

4.2. Evaluation of different recycling rates

Table 3 shows a minimum at 90% recycling rate for energy use and NO_x emissions. The minimum is due to the fact that recycled materials and longer transportation distances have less impact than extraction and refining of virgin materials. At recycling rates greater than 90%, local transport for emptying collection boxes and delivery of batteries to sorting plants increases rapidly.

The use of renewable energy remains constant at increasing recycling rates but non-renewable energy use decreases by 5% when increasing the recycling rate from 25 to 90%. The difference between 25 and 90% recycling is a decrease of approximately 87% for cadmium and nickel resource use and emissions. CO_2 and SO_x emissions decrease by 30 and 80%, respectively. Extraction and refining of virgin nickel give rise to high SO_x emission, which decrease significantly at higher recycling rates.

The total NO_x emission decreases by 39% when comparing no recycling with 90% recycling (Fig. 2.). The contribution of transportation and sorting increases from 7.5 to 53%. The minimum total NO_x emission is found at a 90% recycling rate since it is modelled that increased local truck transportation for collection is needed to achieve very high collection rates. When no batteries are recycled, battery materials contribute most to the total NO_x emission (35%) followed by battery manufacturing Cd and mining/refining. At recycling rates greater than 80%, the recycling process leads to a net reduction in NO_x emission due to higher efficiency than the alternative use of liquid petroleum gas. At different recycling rates, the recycling activity contributes to 4 to -1% of the total NO_x emissions. No significant

Table 3
Selected inventory data for the NiCd battery life cycle (excl. user phase) for different end-of-life treatment methods

	Landfills 100%	Incineration 60%/landfills 40%	Recycling 90%/incineration 6%/landfills 4%	Recycling 100%
Renewable energy (MJ/W h)	0.16	0.16	0.14	0.14
Non-renew. energy (MJ/W h)	5.18	5.15	4.29	4.32
CO_2 [a] (kg/W h)	0.41	0.41	0.26	0.26
NO_x (g/W h)	0.56	0.56	0.34	0.47
SO_x (g/W h)	5.45	5.45	0.83	0.32
Cd (resource) (g/W h)	4.1	4.1	0.41	0
Ni (resource) (g/W h)	5.1	5.1	0.51	0

[a] Carbon from fossil sources.

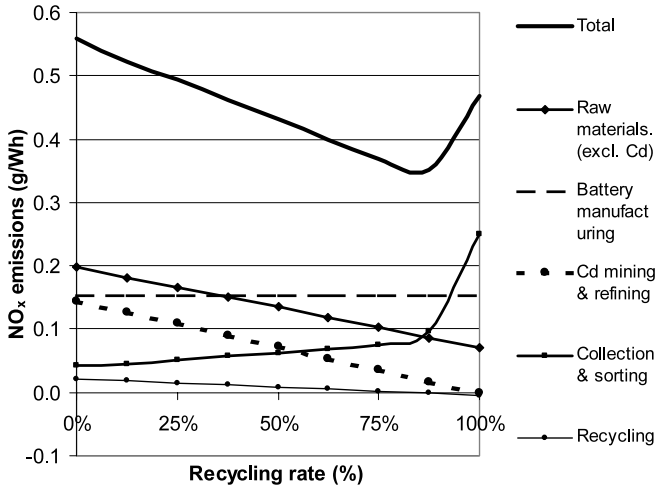

Fig. 2. NO_x emission from the NiCd battery life cycle (excl. user phase) at different recycling rates.

differences were found among the analysed data categories if other energy sources (natural gas, hydropower and coal power) were used in the recycling process instead of liquid petroleum gas.

Fig. 3 shows that an increase in recycling rate from 0 to 90% decreases the total primary energy use by 17%. As a percentage of the total energy use, collection and sorting energy increases from 0.6 to 5%, while energy use in raw materials production decreases from 36 to 15%. By using recycled metals, the energy for the

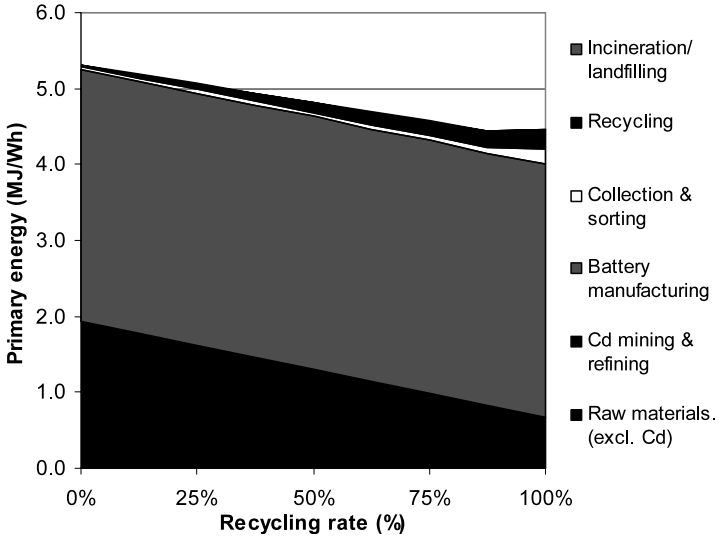

Fig. 3. Primary energy use of the NiCd battery life cycle (excl. user phase) at different recycling rates.

processing of battery raw materials is reduced by 65% compared with virgin materials only. Energy use in the battery manufacturing activity remains constant irrespective of the recycling rate.

The way of electricity generation may alter the absolute values of primary energy use. If all electricity is generated by hydropower, the total primary energy is 2.1–2.8 MJ/W h and NO_x emission 0.17–0.31 g/W h. Corresponding values for coal power is 8.4–10 MJ/W h primary energy and 91–107 g NO_x/W h. Compared to the country specific electricity mix, primary energy use is reduced by half or doubled depending on the energy conversion efficiencies of the different power sources.

To quantify the energy savings made by using secondary materials, the total energy required for recycling must be allocated between the nickel and cadmium recovered. Allocation on a mass basis is preferable to economic allocation since physical parameters are constant (ISO, 1998). Economic values of recovered metals fluctuate over time and do not follow open market prices since the Swedish battery fund pays for the recycling of NiCd batteries. Considering primary energy for collection and recycling of 1 kg of cadmium (mass allocation Cd/Ni = 44/56), 19 MJ is needed for transportation and 19 MJ for recycling. Corresponding values for 1 kg nickel are 20 MJ for collection and 20 MJ for recycling. Compared with extraction and refining of virgin metal, 54 and 75% less primary energy is needed to recycle cadmium and nickel, respectively. The corresponding values using economic allocation (metal prices US$98/kg, Cd/Ni = 7.6/92 (USGS, 1999)) are decreases of 90 and 59%, respectively.

4.3. Metal emissions in short- and long-term perspectives

Total metal emission from different activities is shown in Table 4. At a 25% recycling rate, 0.23–0.26% of the potential metal emission occurs in a 100-year perspective. Most of the metal emission (96–98%) to water occurs in the incineration and landfilling activities, irrespective of whether short- or long-term emissions are considered. Two percent of the cadmium and nickel emission to water occur during battery manufacturing. The emission of cadmium to air takes place mainly (99%) in the incineration and landfilling activities. For nickel, 82% of the emission occurs during incineration and landfilling, while 15% originates from raw materials extraction.

A comparison of metal emissions over the NiCd battery life cycle, considering different methods of end-of-life treatment, shows that the greatest nickel emission to water occurs for 100% landfilling (Fig. 4). The nickel emission to water decreases by 52%, when comparing landfilling with incineration/landfilling. The reduction in emissions for incinerated batteries is due to the fact that metals are more stable in the ash after the addition of sulphides.

The emission of cadmium to air is greatest for incineration/landfilling. Comparing incineration/landfilling with recycling and landfilling, shows that the cadmium emission to air decreases by 75 and 98%, respectively. The nickel emission to air shows the same trends as for cadmium air emission.

Table 4
Emissions of nickel and cadmium to air and water throughout the NiCd battery life cycle (excl. user phase)

	Total	Raw material (excl. Cd) (%)	Cd extraction/refining (%)	Battery manufacture (%)	Collection and sorting (%)	Recycling process (%)	Incineration and landfills (%)
$Cd_{(aq)}$ (mg/W h)	1.4	0.0	1.9	2.0	0.0	0.1	96.0
$Ni_{(aq)}$ (mg/W h)	9.2	0.0	0.0	1.8	0.0	0.1	98.1
$Cd_{(air)}$ (mg/W h)	9.4	0.1	0.3	0.0	0.0	0.1	99.4
$Ni_{(air)}$ (mg/W h)	2.8	15.1	0.4	2.0	0.0	0.1	82.4

End-of-life: 25% battery recycling; 30% landfilling and 45% incineration; short-term metal emissions from landfills.

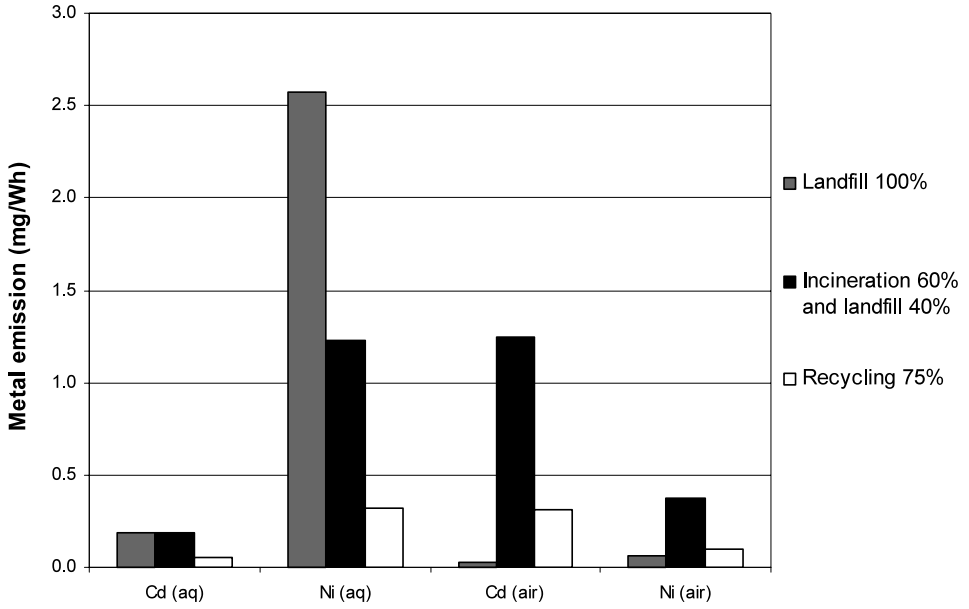

Fig. 4. Total emissions of nickel and cadmium throughout the NiCd battery life cycle (excl. user phase) for different methods of end-of-life treatment (short-term perspective).

Incineration plants with extensive systems for flue gas cleaning lead to low emissions of metals to air and water. Uncontrolled incineration may, however, volatilise cadmium thereby increasing its mobility. Fires in landfills may cause considerable emissions of metals, but this scenario has not been considered in this study. The addition of organic sulphides to ash stabilises metals, but oxidising conditions can make sulphides soluble in water. The mobility of metals may also increase in acidic environments.

Of the total nickel and cadmium content in batteries, 0.08–0.52% of the metals is released to air and water in a 100-year perspective. The potential cadmium and nickel emissions in a long-term perspective are 4.1 g Cd/W h and 5.1 g Ni/W h, i.e. approximately 300–400 times greater than in the short-term perspective. It is, however, not possible to determine the distribution of different metal species and their potential bioavailability.

5. Environmental impact during the NiCd battery life cycle

Primary energy use and the emission of CO_2 are most significant during battery manufacturing, and significant SO_x and NO_x emissions arise in the extraction and refining of raw materials. At the estimated 25% recycling rate in Sweden, emissions and resource consumption of metals are significant in the end-of-life treatment activities.

Depending on the type of battery application, energy in the user phase of the battery and charger losses may contribute to the most significant energy use (and environmental impact) of the NiCd battery life cycle. Different user patterns for a 1 W h NiCd battery (e.g. cycle life 10–1000 cycles and charger efficiency 0.2–0.8) correspond to 0.045–18 MJ electrical energy (0.11–45 MJ primary energy at a conversion efficiency of 0.4). The total primary energy use in the production of NiCd batteries (4.4–5.3 MJ/W h) and battery charging is in the range 4.5–50 MJ/W h NiCd battery. Primary energy in the user phase may account for 2.0–91% of the energy during the NiCd battery life cycle.

Jensen and Petersen, (1999) concluded that the weighted potential environmental impacts were significant regarding energy use in stand-by mode of the charger. The weighted environmental impact was greater for the production of the battery charger than for batteries. When comparing the environmental impact of other rechargeable battery technologies (lead–acid, nickel–metal hydride and lithium–cobalt) with NiCd batteries, no significant differences were found apart from the higher toxicity of NiCd batteries (Kertes, 1996; Törnblom, 1996; Jensen and Petersen, 1999). The primary energy use for the manufacture of different battery types has been calculated to be 5.9 MJ/W h (NiCd), 3.3 MJ/W h (nickel–metal hydride) and 0.40 MJ/W h (non-rechargeable alkaline manganese) (Kirmayer, 1995). The differences depend mainly on the varying energy densities of the battery systems. Refining of raw materials and manufacturing of one alkaline–manganese cell requires 3–4 times less energy than equivalent NiCd or NiMH cells (Kirmayer, 1995).

Quantification of primary energy requirements for recycled metals relies on estimates and the values may vary depending on the system boundaries chosen. Lankey (1998) estimated the energy required for manufacture of batteries with recycled materials to be approximately half the energy needed to manufacture batteries using only primary materials. In this study, the energy reduction was calculated to be 16%. Lankey (1998) claims that 190 MJ/kg is needed for virgin cadmium production and 22 MJ/kg for recycled cadmium. However, these data are difficult to assess since the allocation principles and the use of different energy carriers were not explained.

Longer transportation distances and the use of recycled materials are compensated for by lower emissions in the raw material extraction and refining activities. Although the transport sector contributes significantly to the environmental impact in society, the transportation distance involved in the collection and recycling of materials is generally of very low environmental significance compared with other kinds of transportation (Hunhammar, 1995). The public awareness of transport is higher than for other sectors since transport is associated with high monetary costs and causes emission in urban areas. The LCI model shows that collection and recycling is beneficial from an environmental perspective, even at very low recycling rates. At high recycling rates, increased NO_x emission in urban areas must be compared with reductions in other emissions and resource use. Since the recycling process is not affected by contaminants, the optimum recycling rate for battery metals tends to be close to a hypothetical 100% recycling rate. At closed loop

recycling, the cadmium recovery efficiency is 0.9997 mainly due a fraction of cadmium lost to nickel-iron scrap. Low overall metal losses are crucial for maintaining a stock of metals in batteries and metal resource availability (Andersson and Råde, 2000).

Uncertainties in the results depend on the choice of methodology and data source. Choices in methodology that could affect the results are, modelling of cadmium and nickel as closed-loop recycling, recycling of steel, choice of model for electricity production and the use of system expansion to allocate the useful waste heat from battery recycling. Uncertain data values include assumptions about metal emissions during battery manufacturing, load factor of trucks and transport distances. Sensitivity analyses have, however, shown that these parameters are of minor importance in the final result. The absolute values may be distorted by methodological choices and data values but the identified trends will remain the same.

By relating the use of portable NiCd batteries in Sweden (1998, 25% recycling) to societal material and energy flows the significance of batteries may be identified. Regarding the amount of MSW in Sweden in 1998, all NiCd batteries sold in 1998 would contribute 40–80 ppm to the total mass of MSW (SCB, 2000). In the case of primary energy, the total energy use during the life cycle (excl. user phase) of portable NiCd batteries in Sweden in 1998 contributed 15–23 ppb to the total primary energy turnover that same year. In a study of the material flows of metals in the municipality of Stockholm in 1995, it was estimated that portable NiCd batteries contributed 91% to the annual metal inflow (total 8.8 ton Cd) and more than 25% of the accumulated cadmium in goods and constructions (120 ton) during the period 1900–1995 (Lohm et al., 1997).

The total emission of cadmium from different sources in Sweden in 1995 were estimated to be 1.8 ton cadmium to water ($Cd_{(aq)}$) (39% from mining residue deposits, 31% from pulp and paper) and 0.78 ton cadmium to air ($Cd_{(air)}$) (42% incineration excl. waste, 22% metal smelters) (SCB, 2000). In this study, (25% recycling 1998), the cadmium emission from NiCd batteries was estimated to be 9.2 kg $Cd_{(aq)}$ and 60 kg $Cd_{(air)}$ during 100 years (cumulative metal emission of batteries before 1998 were not considered). Although the emission of cadmium from NiCd batteries is very low in the short-term perspective, continuous accumulation implies an increase of future emissions. The relative contribution of NiCd batteries to the total dissipative losses will probably increase in the future since cadmium use in other products is declining. Guinée et al. (1999) studied metal flows in the Netherlands and estimated that the cadmium emission from accumulated cadmium in the technosphere would increase by 30% from 1990 until a hypothetical steady state is reached. Accumulation in soil could lead to the exceeding of critical levels for human, terrestrial and aquatic ecotoxicity.

The comparison above shows that there is major concern related to the high toxicity of cadmium and the increasing contribution of NiCd batteries to future potential cadmium emission. Incineration plants and landfills in Sweden emit low levels of metals in the short-term perspective and the main problem is whether spent batteries end up in less controlled waste treatment systems or in the environment.

Although not all metals are released at the same time from landfills, continued dissipative losses give rise to elevated background concentrations. To be able to make a risk assessment, the mobility of cadmium and other metals and human exposure to them in different environments would have to be evaluated.

6. Materials management of cadmium

The primary extraction of most metals is still increasing, and a higher degree of closed-loop recycling of metals could lead to a higher accumulation of metals in the technosphere. The emission from metals accumulated in the technosphere could increase in the future for example through corrosion and inadequately controlled incineration.

Voet et al. (1994) studied material flows of cadmium within the EU and evaluated the effects of different policies for controlling cadmium. They concluded that the proposed EU policy based on (1) end-of-pipe measures (2) phasing out of certain applications and (3) recycling of batteries and certain other products, does not appear to offer a sustainable solution to the dispersion of cadmium. This is primarily due to the inelastic nature of the cadmium supply since cadmium is extracted as a by-product of zinc production. On the inflow side, measures should not focus on cadmium but on the main products zinc and phosphate. Otherwise, recycling may only lead to increased accumulation of cadmium and thus to larger future losses to the environment. On the outflow side, the collection of waste and products containing cadmium and subsequent safe storage in a concentrated form seems to offer the best solution.

Although the results of this study show that increased recycling of NiCd batteries decreases the environmental impact, the end use of recycled cadmium must be considered. If recycled cadmium is used in new portable NiCd batteries, it is uncertain whether the battery will be collected again. Some important reasons why it is difficult to achieve high collection rates are that portable NiCd batteries are dispersed among many battery owners, the usage of batteries in diverse applications, the small size of battery units and low economic value of products (Rydh, 2001).

In order to create incentives for battery recycling a demand for spent materials must be created. A combination of historically low prices, limited growth in the use of cadmium metal and pending environmental legislation has made the use of cadmium uncertain. Zinc miners who produce cadmium as a by-product are now regarding the metal as a cost rather than an asset (Mining Journal, 1998). The market is already oversupplied, and new zinc mines under development in Australia and North America will exacerbate the situation. On the global level, the production of refined cadmium was 19.6 kton in 1998 (Plachy, 2000). On the demand side, cadmium is becoming a one-use metal, and approximately 70% is used in the manufacture of NiCd batteries (Mining Journal, 1998). The major share (75%, 10 kton Cd) of cadmium is used in the production of portable NiCd batteries, while the rest (3.4 kton Cd) is used in industrial NiCd batteries (Plachy, 2000). The price

of portable batteries is 2–10 times greater than for the same capacity of industrial NiCd batteries, which makes the market for portable batteries more profitable.

Recycled cadmium is more expensive than primary cadmium delivered from zinc smelters (Mattsson, 2000). The annual average price of cadmium on the open market has decreased from US$4.05/kg in 1995 to US$0.62/kg in 1998 (USGS, 1999). In 1999, the average nickel price was US$6.16/kg (USGS, 1999). The nickel content of NiCd batteries accounts for the largest contribution to the value of scrapped NiCd batteries.

Material and energy resources generally have lower economic value than labour costs. Since recycling is more labour intensive than landfilling or waste incineration, it is a more expensive way of treating end products. Shapek (1996) studied the economic aspects of battery collection in Florida and concluded that the cost increased with higher collection rates. Greene (1995) evaluated actual and hypothetical scenarios for household battery collection and concluded that neither was cost effective. However, it was not possible to determine the economic benefits of reduced metal contamination.

To make recycled materials competitive with virgin materials measures can aim at increasing consumer demand for recycled material. Examples of such measures are introduction of minimum recycled content specifications, taxes on virgin material and subsidies for products containing recycled material. Another way is to increase the supply of available material for recycling. This can be done by modifying consumer behaviour by introducing e.g. compulsory take back, deposits, fixed target recycling and landfill bans. MacDonagh-Dumler (2000) argues for that supply-increasing policies are the most effective for portable NiCd batteries since the last user does not have sufficient incentive to return batteries to the collection system. The motivation to control toxic material fate is driven by public and environmental health concern, not resource efficiency and economic asset value maximization.

Several actions have been introduced or proposed to increase collection rate e.g. public information campaigns, import fees, refunds on batteries, labelling, disposal charges and improved statistics on battery stocks and flows (Ayres and Ayres, 1996; Environment directorate, 1999). A number of stakeholders (EPBA, 1999; Sempels, 1999) are opposed to the regulation of the battery market, and the implementation of stricter measures to encourage improved collection may thus be delayed. However, agreements implemented on the international market (e.g. extended producer responsibility) may diminish trade distortions.

The above-mentioned aspects make it less probable that high collection rates will be achieved for portable NiCd batteries. Toxic metals may instead be used in products with higher metal content and thus of higher economic value per kg recovered battery (e.g. stationary NiCd batteries), which are more likely to be recycled than portable NiCd batteries. Portable NiCd batteries have their niche in power tools and in emergency lightning due to their high power density and good operational characteristics at high temperatures. New battery technologies (NiMH, Li-ion) containing less toxic materials can replace portable NiCd batteries in most applications (Noréus, 2000). NiMH batteries have not entered the NiCd market

niches since the main development goal has been to achieve high energy density by using foamed positive electrodes. However, higher power density in NiMH batteries can be achieved by using sintered positive electrodes. Another reason may be that some NiMH producers also manufacture NiCd batteries and therefore want to conserve the NiCd market niche (Noréus, 2000).

Uncertainties about future emissions and the mobility of metals for different products mean that it is important to perform further studies on emission factors to different media. An environmental assessment of the recycling of portable zinc–carbon and alkaline–manganese batteries would also be of interest to assess the possibility of decreasing the primary extraction of zinc.

7. Conclusions

Primary energy use and emission of CO_2 are most significant in battery manufacturing. Emissions and resource consumption of metals are significant in the end-of-life treatment, regardless of whether short- or long-term emissions are considered. Transportation for the collection of spent NiCd batteries has no significant environmental impact and thus NiCd batteries can be transported long distances for recycling and it would still be beneficial from an environmental perspective. From an environmental perspective the optimum recycling rate for NiCd batteries tends to be close to 100%.

Batteries manufactured with recycled cadmium and nickel have 16% lower primary energy requirements than if only virgin metals are used. Using recycled cadmium and nickel requires 46 and 75% less primary energy respectively, compared with extraction and refining of virgin metal.

There are considerable uncertainties associated with emissions of metals that may occur in the future. The potential cadmium and nickel emissions were 300–400 times greater than in a 100-year perspective. To avoid dissipative losses, cadmium should be used in products that will probably be collected at the end of their life.

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